

# Marine Turtle Newsletter

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Photo of anthropogenic debris (>3200 pieces) found in the large intestine of a small juvenile green turtle that was found stranded in southern Brazil (see pages 6-8; photo: G.D. Stahelin).

## Articles

- Editorial: Does Delayed Mortality Occur in Sea Turtles That Aspirate Salt Water into Their Lungs During Forced Submergence or Cold Stunning?.....**CW Caillouet, Jr.**
- Foraging by Immature Hawksbill Sea Turtles at Brazilian Islands .....**MC Proietti et al.**
- Case Report: Ingestion of a Massive Amount of Debris by a Green Turtle in Southern Brazil.....**GD Stahelin et al.**
- First Record of the Turtle Barnacle *Stephanolepas muricata* from the Pacific Coast of Costa Rica.....**N Corrales-Gómez & Á Herrera-Ulloa**
- Displacement and Site Fidelity of Rehabilitated Immature Kemp's Ridley Sea Turtles.....**H Lyn et al.**
- Effects of Anthropogenic Activities on Nesting Beaches along the Mombasa-Kilifi Shoreline, Kenya.....**SM Mathenge et al.**
- New Northern Limit of Nesting of *Lepidochelys olivacea* in the East Atlantic Ocean: North Senegal.....**J Fretey et al.**
- From Suriname to Ceará: Green Turtle Found Dead on the Coast of Ceará, Brazil.....**EHSM Lima et al.**

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# Editorial: Does Delayed Mortality Occur in Sea Turtles That Aspirate Seawater into Their Lungs During Forced Submergence or Cold Stunning?

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A Weather Channel episode involving a surfer's rescue and resuscitation after nearly drowning led to this editorial. The on-site medic warned rescuers that the surfer could still be at risk if he had inhaled (aspirated) seawater into his lungs. This prompted me to conduct an Internet and literature search for information relevant to two working hypotheses: (1) sea turtles aspirate seawater into their lungs during forced submergence associated with incidental or directed capture or cold-stunning, and (2) this leads to delayed mortality. I also consulted Jeanette Wyneken, Charles Innis, Brian Stacy, and Craig Harms concerning their research relevant to these hypotheses (personal communication, June-August 2012). Their input was helpful and greatly appreciated. For example, I learned from Charles Innis that seawater aspiration occurs in cold-stunned sea turtles (Stockman *et al. in press*), and Craig Harms alerted me to clarifications concerning definitions of drowning (<http://circ.ahajournals.org/content/108/20/2565.full>).

According to Edmonds (1998), delayed death "*occurs when the [human] victim appears to recover from the [nearly drowning] incident, but then proceeds to die.*" My brief search of the Internet produced the following additional descriptions of effects of aspiration of seawater by humans (see also Lunetta & Modell 2005):

- (1) <http://medical-dictionary.thefreedictionary.com/drowning> - "*Sea water aspiration results in fluid-filled but perfused alveoli, accompanied by a V/Q abnormality due to pulmonary edema; the shifts of fluids and electrolytes in salt water drowning result in hemoconcentration, CHF [congestive heart failure], and hypernatremia.*"
- (2) <http://armymedical.tpub.com/MD0587/MD05870088.htm> - "*The effect on the casualty is different when the incident occurs in salt water. Salt water entering the lungs has a higher solute concentration than the plasma in the bloodstream. This causes fluid to be drawn out of the bloodstream into the lungs, causing a massive pulmonary edema (congestion of the lungs). The concentration of salt in the sea water has drawn the normal body water into the lungs. If enough fluid has been drawn out of the patient's bloodstream, the person may go into shock and drown in his own interstitial fluid (fluid bathing the cells).*"
- (3) <http://emedicine.medscape.com/article/772753-overview> - "*Additional classification may include the type of water in which the submersion occurred, such as freshwater and saltwater, or natural bodies of water versus man made. Although initial treatment of submersion victims is not affected by the type of water, serum electrolyte derangements may be related to the salinity of the water (particularly if large amounts of water are ingested), while long-term infectious complications are primarily related to whether the victim was submersed in a natural or a man-made body of water.*"

The sea turtle pulmonary system consists of glottis, trachea, a bronchus to each lung, and left and right lungs (Wyneken 2001, 2006). When a sea turtle surfaces and dives under normal

circumstances, its glottis opens at surfacing to allow air passage into the lungs and it is closed during normal breath-hold diving; also, the anterior tissue lining the nares is erectile in adult sea turtles, and has the ability to seal the nostrils when the turtles are submerged (Wyneken 2001, 2006).

Physiological effects of normal, quiescent, breath-hold dives by sea turtles are mild to moderate compared to those associated with vigorous breath-hold swimming, or with struggling accompanying forced submergence (Stabenau *et al.* 1991; Moon & Stabenau 1996; Lutcavage & Lutz 1997; Lutcavage *et al.* 1997; Lutz 1997; Hoopes *et al.* 2000; Moon & Foerster 2001; Harms *et al.* 2003; Stabenau & Vietti 2003; Wyneken *et al.* 2006; Snoddy *et al.* 2009; Work & Balazs 2010). Voluntary dives appear aerobic, with little if any increase in blood lactate and only minor changes in acid-base balance; however, during such dives, a sea turtle incurs an oxygen debt, which must be repaid through resumed breathing when it surfaces (Lutcavage & Lutz 1997). During voluntary dives, sea turtles typically do not exceed their aerobic diving limit (Southwood *et al.* 1999; Hochscheid *et al.* 2007). However, during vigorous breath-hold swimming, or struggling accompanying forced submergence, oxygen stores are consumed rapidly, anaerobic glycolysis occurs, and acid-base balance is disturbed, sometimes to lethal levels (Lutcavage & Lutz 1997).

Apparent recovery of comatose or debilitated sea turtles collected and resuscitated following forced submergence can require many hours (Balazs 1986; Harms *et al.* 2003; Snoddy *et al.* 2009). If seawater aspiration by sea turtles is a common occurrence during forced submergence, 24 hr of post-submergence treatment may not be sufficient to assure their survival following release (<http://www.nefsc.noaa.gov/nefsc/publications/crd/crd1110>). For example, of nine Kemp's ridleys (*Lepidochelys kempii*) found stranded alive in 1994, only two survived and were released; five died within 72 hr of being retrieved for rehabilitation, and two survived more than 72 hr before dying (Cannon 1998). Of the two that survived more than 72 hr before dying, necropsies revealed that "... both had abnormal development of the lungs", and that "[d]efects causing inefficient gas exchange were likely contributors to the demise of these two animals" (Cannon 1998).

It is difficult to confirm drowning as the cause of death due to forced submergence based on necropsies of dead-stranded sea turtles (Wolke & George 1981). NMFS (2012) provided the following description: "*Suspicion of drowning (i.e., involuntary or forced submergence) in a stranded sea turtle, such as that for PTT102741, is not based on any one finding alone (e.g., sediment in the lungs), but relies on exclusion (to the extent possible) of other potential causes of death/debilitation. It is not unusual to identify drowning as a potential cause of death for any air-breathing animal that lives in an aquatic environment. Furthermore, there is nothing diagnostic about a forced submergence scenario that would necessarily distinguish it from other causes of drowning.*"

*A conclusion of forced submergence is generally based on other findings, specifically the lack of any significant trauma, disease or indicators of poor general health; evidence of a sudden event (e.g., food in the mouth or esophagus); and/or absence of harmful algal bloom and other toxins.”* Interestingly, various commentaries on preliminary results of Brian Stacy’s necropsies of sea turtles found stranded in the northern Gulf of Mexico during 2011 mentioned evidence of drowning, including aspiration of sediment-rich water. That sediment-rich water most likely was seawater.

Incidental capture of sea turtles in towed trawls or dredges represents a special case of forced submergence. When sea turtles are caught in shrimp trawls, which are towed at rates of 0.5-1.5 m s<sup>-1</sup> (<http://www.fao.org/fishery/fishtech/1021/en>), they initially face the current and are pressed against the netting above them (e.g., Ogren *et al.* 1977; [http://www.nmfs.noaa.gov/pr/species/turtles/loggerhead\\_video.htm](http://www.nmfs.noaa.gov/pr/species/turtles/loggerhead_video.htm)). Any sea turtle caught incidentally in a shrimp trawl obviously failed to out-swim or out-manuever the trawl. Therefore, the question arises whether sea turtles caught in towed trawls aspirate seawater due to exposure to currents and turbulence within the trawls. A somewhat comparable human model might be the seawater aspiration syndrome in divers, including those affected by a fast-towed underwater search, faulty (leaking) SCUBA equipment, exertion, swimming against currents, exhaustion, panic, etc. that can lead to seawater aspiration as part of the sequence leading to their death (Edmonds 1998; Lunetta & Modell 2005). Van Beeck *et al.* (2005) proposed a new definition of drowning in humans, noting large variations related to types of activity and water to which they are exposed (including oceans).

If delayed mortality due to seawater aspiration occurs in sea turtles, it might help explain continued strandings of sea turtles concomitant with shrimp trawling in the southeastern U.S., despite regulations requiring turtle excluder devices (TEDs) in shrimp trawls (Stabenau *et al.* 1991; Caillouet *et al.* 1996; Shaver 1998; McDaniel *et al.* 2000; Epperly 2003; Lewison *et al.* 2003; Sasso & Epperly 2006; NMFS 2012). Obviously, prolonged forced submergence and associated mortality in sea turtles could result from various violations of TED regulations described by NMFS ([http://www.nmfs.noaa.gov/pr/pdfs/species/deis\\_seaturtle\\_shrimp\\_fisheries\\_interactions.pdf](http://www.nmfs.noaa.gov/pr/pdfs/species/deis_seaturtle_shrimp_fisheries_interactions.pdf)): “*NMFS has recently noticed compliance issues with TED requirements in the shrimp fisheries. During numerous evaluations conducted in both the Gulf of Mexico and Atlantic Ocean over the past two years, NMFS gear experts have noted a variety of compliance issues ranging from lack of TED use, TEDs sewn shut, TEDs installed improperly, and TEDs being manufactured that do not comply with regulatory requirements.*” When sea turtles aspirate seawater into their lungs during forced submergence and cold-stunning, delayed mortality might occur even though the turtles are resuscitated (Balazs 1986; Norton 2005; Wyneken *et al.* 2006; NMFS SEFSC 2008; Canion & Rogers 2010a, 2010 b), provided medical treatment, and released alive ([http://www.nero.noaa.gov/prot\\_res/stranding/SeaTurtleHandlingResuscitationv1.pdf](http://www.nero.noaa.gov/prot_res/stranding/SeaTurtleHandlingResuscitationv1.pdf)). Evidence of or speculation about seawater inhalation into the lungs of sea turtles and its consequences can be found in Ryder *et al.* (2006), Wyneken *et al.* (2006), Innis *et al.* (2009), Snoddy & Williard (2010), Work & Balazs (2010), Upite (2011), and Stockman *et al.* (*in press*). The following description of delayed effects of seawater aspiration by sea turtles was given by Wyneken *et al.* (2006): “*Saltwater drowning is a serious condition. Even if the patient is*

*resuscitated, the residual seawater in the lungs induces a secondary drowning as body water follows the osmotic gradient, leaving the pulmonary tissue and entering the lungs. Treatment is difficult, and the prognosis is grave. Antibiotics, fluids, positional drainage (inclined with head down), suction, and oxygen supplementation may be necessary.*”

Sea turtles that are comatose, lethargic, or active after known exposure to forced submergence and cold-stunning, but otherwise appear healthy, are the best candidates for further research on potential effects of seawater aspiration on delayed mortality. However, for research purposes, they should be tracked after release (e.g., Snoddy & Williard 2010), or retained in captivity under conditions amenable to resuscitation, medical treatment, extended observation, and rehabilitation, over periods long enough for full evaluation of their recovery (Upite 2011). If they die while receiving medical treatment, more information would be available on the cause(s) of death. In any case, the working hypotheses concerning delayed mortality associated with seawater aspiration by sea turtles subjected to forced submergence or exposed to hypothermia seem worthy of further attention and research.

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## Foraging by Immature Hawksbill Sea Turtles at Brazilian Islands

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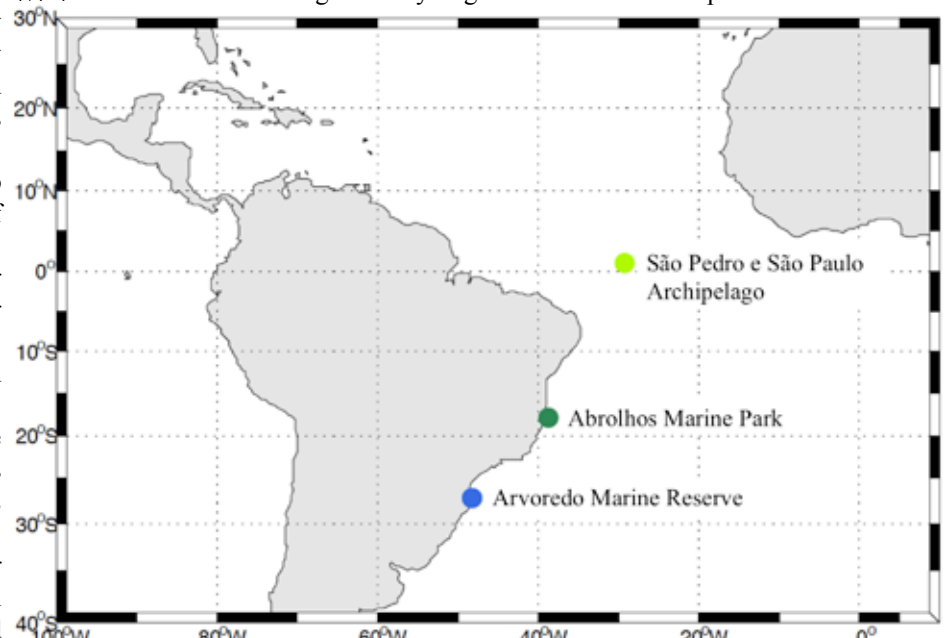
Among sea turtle species, hawksbills (*Eretmochelys imbricata*) have suffered one of the longest and most intense exploitation processes (Mortimer & Donnelly 2008). This species inhabits tropical waters of all oceans and is especially associated with coral reefs due their preferentially spongivorous diet (León & Bjorndal 2002). However, hawksbills may also inhabit other hard-bottomed benthic habitats such as seagrass beds, rocky reefs, mangrove bays, and mud flats (Mortimer & Donnelly 2008). Since there are few studies concerning the ecology and behavior of immature hawksbills at Brazilian feeding areas, this work is essential for understanding hawksbill populations and their ecological roles within their habitats. Here we studied immature hawksbills foraging around three high-biodiversity areas in Brazil (Fig. 1): (1) the São Pedro e São Paulo Archipelago (SPSP), which is over 1,000 km from the coast of Rio Grande do Norte state and has deep rocky shores with high occurrences of hawksbills and green sea turtles (*Chelonia mydas*); (2) the Abrolhos Marine National Park, which is approximately 70 km from Bahia state and has calm shallow reefs commonly visited by hawksbills and greens, and occasionally loggerhead sea turtles (*Caretta caretta*) and; (3) Arvoredo Island, which is the largest island of the Arvoredo Biological Reserve and has high occurrences of green turtles and some hawksbill turtles.

Snorkel and scuba dives were conducted for in-water observations and turtle captures. For each turtle sighting we recorded the date, time, dive location, depth, substrate type, estimated carapace length, behavior (swimming, feeding, resting on the bottom, assisted resting – turtle resting under any structure – and associations with fish) and other relevant characteristics (methods adapted from Houghton *et al.* 2003). We also attempted to photograph behavior and the facial profiles of each sea turtle. When possible, hawksbills were manually captured after recording the sighting. Captured turtles were tagged (Inconel tags provided by Project

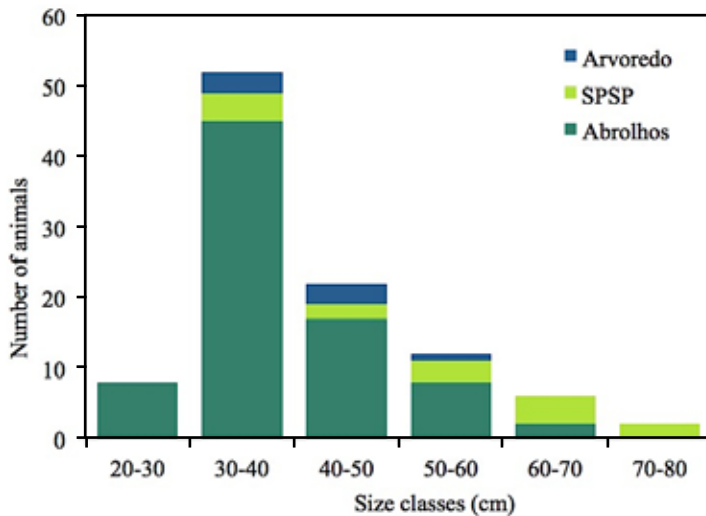
Tamar-ICMBio), weighed, measured (curved carapace length – CCL) and photo-identified (Reisser *et al.* 2008). Epidermis and scute samples were also taken for genetic and isotope analysis. After these procedures were complete, the turtles were immediately released close to their capture locations.

From a total of 80.1 dive hours performed at Abrolhos there were 162 underwater sightings and 65 individual hawksbills captured. At SPSP we dived for 29.2 hours and this resulted in 73 underwater sightings and 12 individuals captured. At Arvoredo Island we performed 235 dive hours with 22 underwater sightings and 6 individuals captured, and one of the turtles was subsequently recaptured twice.

The size of captured turtles ranged from 24.5 – 63.0 cm CCL at Abrolhos (mean = 37.9 cm), 30 – 75 cm at SPSP (mean = 53.7 cm), and 30 – 59.5 cm (mean = 41.3 cm) at Arvoredo (Fig. 2). Mean sizes were significantly larger at SPSP when compared to the other two



**Figure 1.** Hawksbill foraging grounds in Brazil (this study): Arvoredo Marine Reserve, Abrolhos Marine Park and São Pedro e São Paulo (SPSP).



**Figure 2.** Numbers of hawksbill turtles captured in Arvoredo Marine Reserve, Abrolhos Marine Park and São Pedro e São Paulo (SPSP), according to size classes.

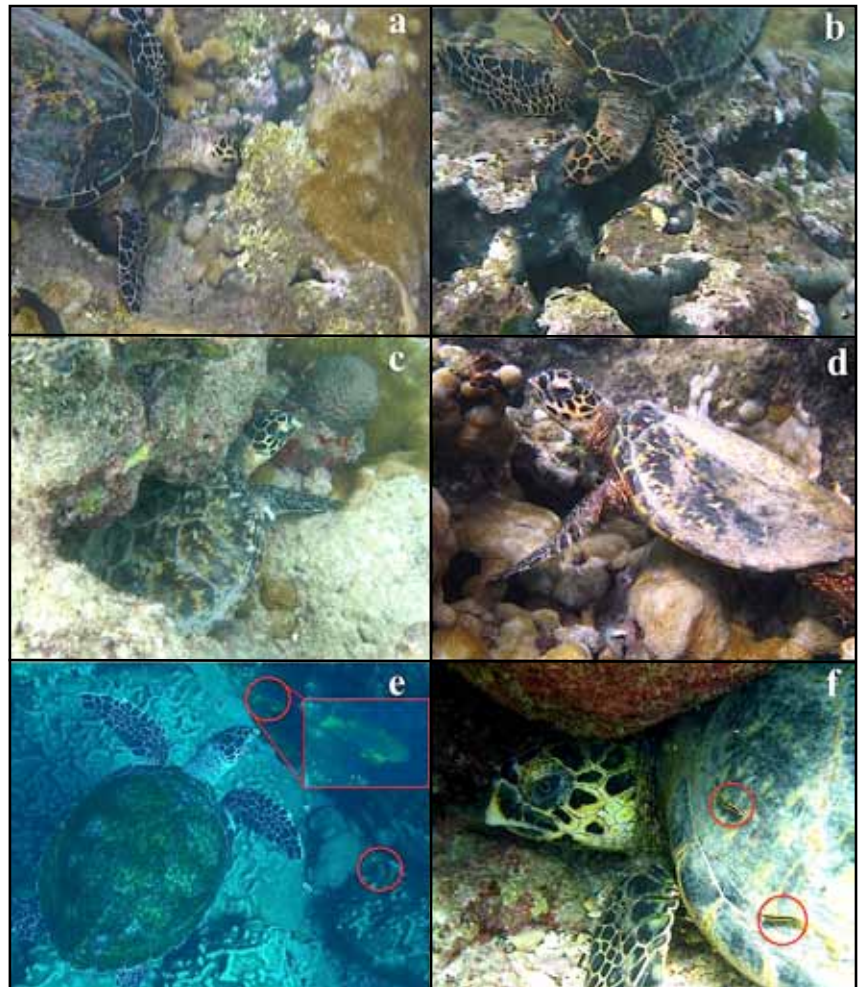
areas ( $p < 0.05$ ), but mean sizes between Abrolhos and Arvoredo did not differ significantly ( $p > 0.05$ ), as demonstrated by a Student's *t*-test. Due to the high abundance of small turtles at Abrolhos Park, we believe that this is an important recruitment area for hawksbill turtles. On the other hand, we observed relatively large size classes at SPSP and this indicates that this area is an important feeding ground for older hawksbills, perhaps due to its proximity to the Caribbean, where the majority of Atlantic hawksbill rookeries are located (Mortimer 2007).

Hawksbill feeding activity was recorded in 28.9% ( $n = 71$ ) of the observations and consistently occurred at shallow portions of the reefs (depths shallower than 4 m) at Abrolhos and Arvoredo, and at greater depths (deeper than 8 m) at SPSP. Feeding occurred throughout the day (observed from 0600 to 1900 hours) and hawksbills seemed to select their prey by searching for them slowly while swimming close to the reef or rocks. In all of the feeding observations hawksbills selected sessile benthic organisms, mainly zoanths (green sea mat, *Zoanthus sociatus*, and white encrusting zoanthid, *Palythoa caribaeorum*) and occasionally sponges. Although most studies on hawksbill diets report a preference for sponges (León & Bjorndal 2002; Meylan 1988), feeding on zoanths has also been observed (Stampar *et al.* 2007).

Resting behavior (20.3% of sightings,  $n = 50$ ) was also observed throughout the day, and hawksbills apparently chose deeper sites for this activity, resting mostly in spots deeper than 4 m at Abrolhos and Arvoredo, and greater than 10 m at SPSP. In 70% ( $n = 35$ ) of resting observations turtles chose spots under rocks, demonstrating a preference towards assisted resting.

The frequency of other observed behaviors was found to be 48% ( $n = 118$ ) swimming and 2.8% ( $n = 7$ ) activity associated with reef fish. Cleaning activity on sea turtles by three reef fish species was recorded. There were four sightings at Abrolhos of cleaner fish (yellow line goby, *Elacatinus figaro*) nipping at the turtle's carapace, with up to three fish cleaning simultaneously. There were two sightings at SPSP of the endemic Saint Paul's gregory (*Stegastes sanctipauli*) cleaning the neck and carapace of a turtle and one observation at Arvoredo of a juvenile French angelfish (*Pomacantus paru*) feeding off a carapace. Associations between sea turtles and fish in Brazil have been recorded for many fish species including *P. paru* (Sazima *et al.* 2010), but to our knowledge this is the first record of *E. figaro* and *S. sanctipauli* cleaning hawksbill sea turtles.

By photographing the facial profiles of hawksbills upon initial capture, we were able to recognize some turtles (31 individuals on 52 occasions) through underwater photo-ID (see Figs. 3d and 3f). This demonstrates the great potential of photo-ID for conducting non-intrusive population studies. Intervals between initial capture and posterior "recaptures" (through underwater photo-ID or manual capture) varied from 1 to 242 days at SPSP, 1 to 297 days at Abrolhos, and 367 to 671 days for Arvoredo Island. We believe that additional field surveys would reveal even longer periods of permanency, further highlighting hawksbill residency at these feeding grounds. The permanency of this tropical species at



**Figure 3.** Examples of hawksbill behaviors at the study sites: a/b) feeding on zoanths; c) assisted resting; d) unassisted resting; e) Saint Paul's gregory cleaning at SPSP; f) yellow line goby cleaning at Abrolhos. Red circles in e and f indicate fish locations. Images d and f are examples of typical underwater photo-ID. Photographs by M.C.P.

Arvoredo Island is remarkable considering that this area reaches temperatures as low as 13°C in the winter (pers. obs. in July 2007). This work demonstrates that Brazil hosts important hawksbill turtle foraging grounds, which should be preserved for the recovery of *E. imbricata* populations. Forthcoming stable isotope analyses will provide further understanding of hawksbill diet and habitat use at these Brazilian islands. Genetic studies currently underway will link these foraging populations to their stocks of origin, improving our current knowledge on hawksbill connectivity in the Atlantic Ocean and enhancing our ability to protect this species.

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## Case Report: Ingestion of a Massive Amount of Debris by a Green Turtle (*Chelonia mydas*) in Southern Brazil

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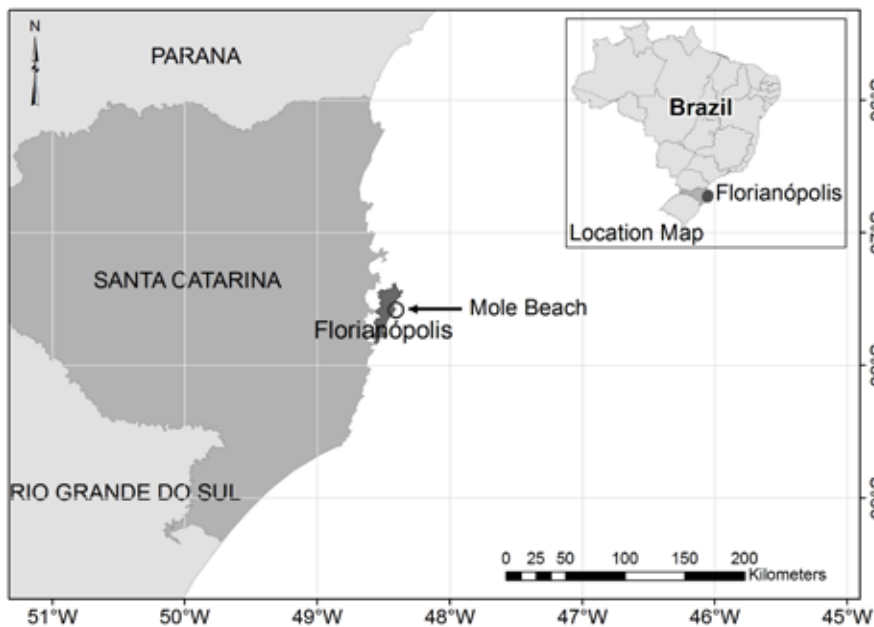
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Marine debris is considered any solid waste (plastic, polystyrene, rubber, foam, glass, metal, cloth, and other man-made materials) that enters the marine or coastal environments from any source (Coe & Rogers 2000). The main sources of marine debris are litter carried into the sea from land-based sources in industrialized and highly populated areas and wastes from ships, fishing and recreational vessels (Derraik 2002). However, regardless of the source, marine debris can have serious ecological and economic consequences. These adverse impacts have been documented all over the world. According to Gregory & Ryan (1997), plastic pollution is estimated to represent between 60% and 80% of the total marine debris in the world’s oceans. Within just a few decades since mass production of plastic products commenced in the 1950s, plastic debris has accumulated in terrestrial environments, in the open ocean, on shorelines and in the deep sea (Barnes *et al.* 2009).

Every year, many species of marine animals, including sea turtles, marine mammals, seabirds and fish die from becoming entangled or ingesting plastic debris (Laist 1987). According to Carr (1987) sea turtles are particularly prone to eating plastics and other floating debris. Juvenile sea turtles are frequently exposed to pollution in convergence zones and most species are exposed in nearshore habitats, where they feed (Bjorndal *et al.* 1994). Evidence indicates that the high occurrence of non-food items in sea turtle species may be related to mistaken ingestion of plastics, due to its similarity to prey items (Plotkin *et al.* 1993), or even to incidental ingestion along with a prey (Tomás *et al.* 2002).

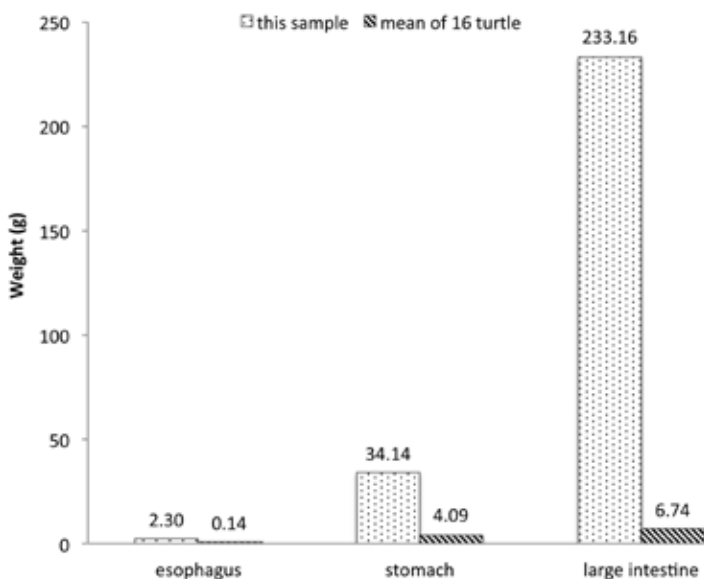
On 18 July, 2010 a juvenile green turtle (*Chelonia mydas*) was rescued by Projeto Tamar (Brazilian sea turtle conservation program) after stranding at Mole Beach, in Florianópolis municipal district, Santa Catarina State, Brazil (Fig. 1). On admission, the





**Figure 1.** The location where the *C. mydas* stranded. Mole Beach is located on the island of Florianópolis, in Santa Catarina State, Brazil.

animal was measured (39 cm curved carapace length, 38 cm curved carapace width), weighed (6 kg), and received a thorough physical examination. The turtle was weak, in poor body condition, malnourished and emaciated. Clinical signs included dehydration, prostration and areflexia. Death occurred a few hours after initial supportive care. In order to determine the cause of death, a necropsy was performed on the individual. During the procedure, the turtle had its sex determined as a male by visual examination of the gonads. All coelomic organs were examined and no apparent gross pathology was noted. However, a massive amount of debris was found in its digestive tract and was apparently blocking food passage. The gastric and intestinal mucosa showed the presence of several ulcers, probably caused by the presence of debris, which could have possibly led to excess gastric acid production. The gut content was



**Figure 2.** Comparative weight of items found in this sample and those found in 16 other turtles at the same area.

then separated according to its location: esophagus, stomach, small and large intestines. Contents were carefully rinsed in a sieve with a 1 mm mesh and marine debris was separated and dried at 50 °C.

Afterwards, the samples were divided into seven categories: soft plastics, hard plastics, nylon, other plastics, latex, textile and other/unknown. Only debris items larger than 5 mm were counted. Any particles smaller than 5 mm were considered fragments of another piece, and were only weighed.

In the esophagus, 18 items were found (total dry weight: 2.30 g), in the stomach there were 308 items (34.14 g), and in the large intestine there were 3,267 items (233.16 g, see cover photo). No anthropogenic debris was found in the small intestine.

It is likely that the obstruction caused by the marine debris ingestion led this individual to death. In terms of comparative data (Fig. 2), this turtle had an enormous amount of garbage in its stomach and large intestine. The mean number of items found in the gastrointestinal tracts of other turtles (16 animals) stranded in the same area was: 9.67 items

± 15 (range: 1 - 27; total dry weight: 0.01 - 0.4 g) in the esophagus; 54.2 ± 50.5 (1 - 136; 0.02 - 16.39 g) in the stomach, 11.4 ± 19.1 (1 - 45; 0.02 - 4.81 g) in the small intestine and 128 ± 182 (6 - 732; 0.08 - 40.92 g) in the large intestine. Additionally, a comparison was made between our results and those obtained in different studies (see Table 1). Our study shows a significantly higher amount of debris than the others, although only one case report is presented here.

Death by plastic ingestion may be caused by reduced stomach capacity (Ryan 1988); obstruction (Lazar & Gracan 2011) or exposure to toxic compounds (Bjorndal *et al.* 1994). According to Laist (1987), starvation is the major cause of death for animals that ingest anthropogenic debris. Nutrient absorption from food takes place as the items pass through the digestive tract. Therefore, in case of a gut blockage, the animal will starve to death. Additionally, even if there is no blockage, consumption of plastics in the place of food items may cause sublethal effects, such as partial obstruction of the gastrointestinal tract and reduction

Sp.	N	Range	Debris	Min. size	Reference
<i>Cc</i>	43	1-59	366	1	Tomás <i>et al.</i> 2002
<i>Cc</i>	19	1-27	82	1	Lazar & Gracan 2011.
<i>Cm</i>	34	3-134	1602	n/a	Tourinho <i>et al.</i> 2010.
<i>Cm</i>	56	n/a	3737	<1	Guebert-Bartholo <i>et al.</i> 2011.
<i>Cm</i>	23	1-29	n/a	n/a	Bugoni <i>et al.</i> 2001.
<i>Cm</i>	1		3593	0.5	Present study

**Table 1.** Incidence and amount of debris in the digestive tracts of sea turtles reported in different studies. Sp = species; *Cc* = loggerhead, *Cm* = green turtle, Range = range of pieces of anthropogenic debris found in the digestive tracts of sea turtles, Debris = total debris found in the digestive tracts of sea turtles. Min. size = minimum size (in cm) of anthropogenic debris considered.

of feeding stimulus (Ryan 1988; Bjorndal *et al.* 1994; McCauley and Bjorndal, 1999). Floating plastic debris are also known to absorb toxic contaminants from surrounding waters, increasing considerably its toxicity when ingested. These contaminants include persistent organic pollutants such as polychlorinated biphenyls (PCBs), dichlorodiphenyldichloroethylene (DDE), nonylphenol and phenanthrene, which can become several orders of magnitude more concentrated on the surface of plastic debris than in the water column (Teuten *et al.* 2009).

Recently, it has been suggested that plastics could transfer harmful chemicals to living organisms (Oehlmann *et al.* 2009; Koch & Calafat 2009). A range of chemicals are used as additives in the manufacture of plastics, such as phthalate plasticizers and brominated flame retardants. These substances are potentially harmful and have been associated with carcinogenic and endocrine disrupting effects (Teuten *et al.* 2009).

Although only one case report is presented in this study, it shows how devastating marine debris can be to marine animals. Further research is required to better understand the impacts of ocean litter on sea turtle survival. Moreover, priority implementation measures should be discussed in order to prevent and reduce marine debris and its impacts on the environment. Efforts to reduce waste, increase recycling, increase use of reusable items, implement education programs and beach clean ups are also important as a means to mitigate the global marine debris problem.

**Acknowledgements.** We thank Wallace J. Nichols for review and suggestions. Projeto TAMAR, a conservation program of the Brazilian Ministry of the Environment, is affiliated with ICMBio (Chico Mendes Institute for Biodiversity Conservation) and is co-managed by Fundação Pró-TAMAR. Data collection was authorized by ICMBio, through special license number 14122, issued by Biodiversity Authorization and Information System (SISBIO).

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# First Record of the Turtle Barnacle *Stephanolepas muricata* from the Pacific Coast of Costa Rica

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The sessile barnacles (Balanomorpha) included in the family Platylepadidae are obligatory symbionts of motile marine animals, with some species occurring solely on turtles, sea snakes, and fish (Newman & Ross 1976; Pfaller *et al.* 2012). Platylepadid barnacles occur partially to fully embedded within the host's tissues - producing external wall elaborations that serve to anchor the barnacle (Badillo 2007; Ross & Frick 2007; Zardus & Balazs 2007). Stomatolepadine barnacles like *Stephanolepas* are characterized by nearly- to fully-encapsulating the shell in host tissue (Ross & Frick 2011). The shell of *S. muricata* is fragile and has a series of sutural elaborations that radiate outwards so as to cross-anchor the animal deep within the dermis of the host tissue (Fig. 1) (Frick *et al.* 2011).

The first report of *S. muricata* came from the skin of a hawksbill (*Eretmochelys imbricata*) turtle captured in the South China Sea, Southeastern Vietnam (Fisher 1886). Subsequent studies have found *S. muricata* on other sea turtle species - including green turtles (*Chelonia mydas*), loggerheads (*Caretta caretta*) and olive ridleys (*Lepidochelys olivacea*) (Badillo 2007; Frick *et al.* 2011). *Stephanolepas* is currently known from turtles in the following regions: Mediterranean-Eastern Atlantic, Indo-West Pacific, Eastern Pacific, Hawaii and the Galapagos Islands (Frick *et al.* 2011).

The first records of *S. muricata* from Baja California and Sinaloa, Mexico in olive ridleys was presented by Frick *et al.* (2011). In this note, we describe the first record of *S. muricata* on the Central Pacific coast of Costa Rica on a hawksbill turtle.

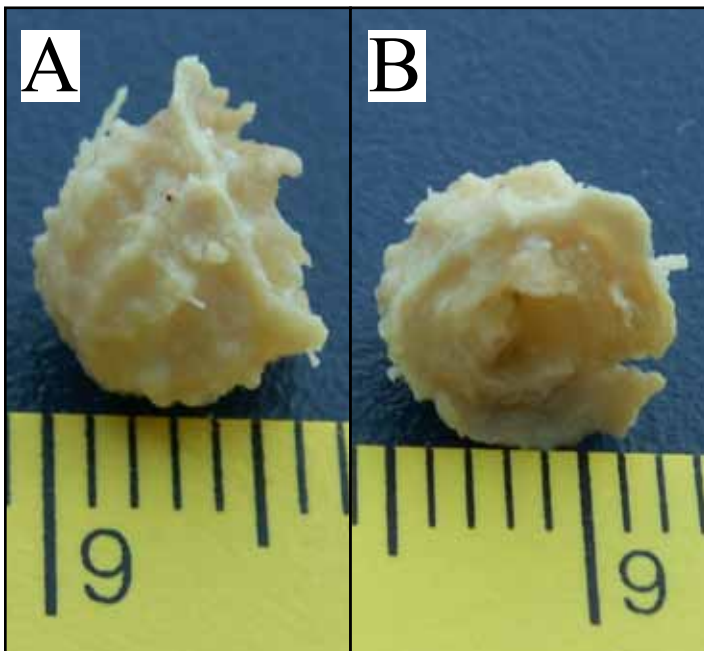
On the night of 22 August, 2008, in the Gulf of Nicoya, Costa Rica, a hawksbill turtle was caught by fishermen near Isla Cedros (Fig. 2) as by-catch from a gillnet. Fishermen removed the turtle from the gillnet and transported it to Parque Marino del Pacífico (Marine Park of the Pacific) where it was admitted for recovery following necessary institutional protocols. The morphometrics of the turtle were 34.6 cm X 29.2 cm (curved carapace length and width), and the turtle's weight was 3.4 kg. Upon arrival, the turtle was examined for epibionts and we collected specimens of the chelonophilic barnacle, *Stephanolepas muricata* Fischer, 1886 (Cirripedia: Coronuloidea: Platylepadidae, Figure 2). Ours is the first report of this symbiotic sea turtle barnacle species from Costa Rica.

The hawksbill turtle admitted to Parque Marino hosted numerous *S. muricata* attached to the leading edges of the front and rear flippers (Figs. 3 & 4), causing deep wounds that altered the normal shape of the flippers. The turtle was placed in fresh water for three days to rehydrate it and to remove epibiota. All barnacles were removed from the turtle's skin thus causing some superficial bleeding. The resulting wounds were treated successfully with topical iodine and silver sulfadiazine cream. On 20 March, 2009, the turtle weighed 6.8 kg and was released near Tortuga Island (9.767183° N, -84.907550° W).

**Acknowledgments.** We thank Cinthya Sancho for helping in treating and healing this sea turtle.

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**Figure 1.** Cirripeds *Stephanolepas muricata* extracted from a hawksbill: A=side view and B=front view.



**Figure 2.** Gulf of Nicoya and Cedros Island, Costa Rica.





**Figure 3.** *Stephanolepas muricata* in the front flipper of a hawksbill.



**Figure 4.** *Stephanolepas muricata* in the rear flipper of a hawksbill.

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## **Displacement and Site Fidelity of Rehabilitated Immature Kemp's Ridley Sea Turtles (*Lepidochelys kempii*)**

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In 2010, numerous immature Kemp's ridley sea turtles (*Lepidochelys kempii*) were incidentally captured by recreational fishermen on piers or stranded live in Mississippi and Alabama and were rehabilitated at the Institute for Marine Mammal Studies (IMMS) in Gulfport, MS. This Critically Endangered sea turtle was once on the brink of extinction, but due to conservation and management efforts on nesting beaches and at foraging grounds, this species is experiencing a population recovery (Crowder & Heppell 2011; Heppell *et al.* 2007). Coastal areas within the Gulf of Mexico represent important developmental habitats for juvenile Kemp's ridleys (Ogren 1989). Immature Kemp's ridleys arrive at these neritic habitats to feed

primarily on crabs and other invertebrates after a transition from their post hatchling pelagic lifestyle (Ogren 1989).

The rehabilitation and release of juvenile and subadult Kemp's ridleys at IMMS presented an opportunity to examine the movements of these poorly understood life history stages in an understudied region of the Kemp's range, the north central Gulf of Mexico. Twelve rehabilitated sea turtles were selected for satellite tracking. During the fall of 2010, six of these turtles were released in Mississippi waters, two miles south of East Ship Island. Due to the high number of sea turtle strandings along the Mississippi coast during the spring of 2011, the other six rehabilitated turtles



Turtle	Release Location	Release Date	Size (cm)	Mass (kg)	Track Days
Squirt	East Ship Island, MS	11/23/2010	30.9*	4.7	117
Crush	East Ship Island, MS	11/20/2010	33.7*	5.7	155
Scuter	East Ship Island, MS	11/23/2010	33.8*	5.9	138
Terry	East Ship Island, MS	11/20/2010	33.0*	5.4	23
Marlin	East Ship Island, MS	11/20/2010	46.1*	12.0	149
Skipper	East Ship Island, MS	11/23/2010	35.8*	6.4	168
Coral	Cedar Keys, FL	4/26/2011	36.4^	6.3	76
Strider	Cedar Keys, FL	4/26/2011	54.5^	20.4	445#
Pearl	Cedar Keys, FL	4/26/2011	35.1^	6.6	57
Oceania	Cedar Keys, FL	4/26/2011	34.5^	6.1	49
Tim	Cedar Keys, FL	4/26/2011	37.5^	7.2	14
Ariel	Cedar Keys, FL	4/26/2011	39.5^	8.4	50

**Table 1.** Lengths, weights and tracking data for the 12 satellite-tagged immature Kemp's ridley sea turtles released in Mississippi and Florida. \*=Straight-line notch-tip carapace length; ^=Curved carapace notch-tip length; #=An active track as of manuscript preparation.

were released near documented immature Kemp's ridley feeding grounds in Cedar Key, Florida (Schmid *et al.* 2003) rather than in Mississippi, in an attempt to prevent re-stranding. The movements of the two groups were compared to examine the possible effects of translocating immature Kemp's ridleys by releasing them in a different location from where they were found. This analysis is presented to provide an initial assessment of site fidelity within the north central Gulf of Mexico.

In the fall of 2010, six rehabilitated immature Kemp's ridleys were released near the Mississippi Sound off Ship Island (N 30° 20.82' W 88° 91.60') (Table 1). In the spring of 2011, an additional six immature Kemp's ridleys were released off Cedar Key, Florida (N 29° 13.335' W 82° 97.76'). The individuals released in 2010 were fitted with a Sirtrack KiwiSat K2G - 202 series platform terminal transmitter (PTT), 371A (n = 3) and 271B (n = 3). The individuals released in 2011 were fitted with a 271B (n = 6) PTT. The battery from each PTT 371A had a lifetime of approximately 115 days at constant power and weighed approximately 170 grams. The battery from the PTT 271B had a battery life of approximately 80 days and weighed approximately 98 grams. Transmitter sizes were consistently less than 3% of each individual's release weight. Each PTT was painted with Tempo Marine, a clear antifouling paint. Prior to application of the transmitter, each turtle's anterior vertebral and costal scutes were sanded and cleaned with acetone. Transmitters were attached following the procedures outlined in Seney *et al.* (2010). Once the epoxy had cured, two coats of the brush-on antifouling paint Interlux Micron were applied to the cured adhesives as well as the non-metal surfaces of the PTT.

Each PTT was set to a duty cycle of 6 hours on followed by 6 hours off to conserve the battery. Messages received from the satellites were processed by CLS America ([www.clsamerica.com](http://www.clsamerica.com)) to give Doppler-derived locations classified by the number of messages used for processing. Location classes included LC 3, 2, 1, 0, A, B, and Z. LC 3, 2, 1, and 0 were derived from a minimum of 4 messages. These classes had estimated accuracies of < 250 m, < 500

m, < 1500 m and > 1500 m respectively. LC A and LC B were calculated from 3 and 2 messages respectively and did not provide accuracy estimation. LC Z indicated an invalid location (Argos 2009).

The Satellite Tracking and Analysis Tool (STAT) (Coyne & Godley 2005) was used to exclude locations in the following categories: 1) LC Z; 2) locations that recorded swimming speeds of 5 km hr<sup>-1</sup> or greater; 3) locations that were recorded at elevations at 0.5 m or greater; and 4) locations that were recorded on dry or over land areas. Incorrect readings (points that crossed land or large areas of water) that were not filtered by STAT were removed manually in ArcMap 9.3. In both release groups the Pearson's correlation coefficients were used to compare the distance from the hooking/stranding and release locations to the time the first transmission was run at an  $\alpha$  level of 0.05. A two-tailed t-test was conducted at an  $\alpha$  level of 0.05 with the average swimming speeds for the two groups. The slopes of the regressions were also analyzed to examine any differences in overall movements. Microsoft® Excel was utilized for these analyses. The time period that was analyzed was constrained to 60 days to reduce

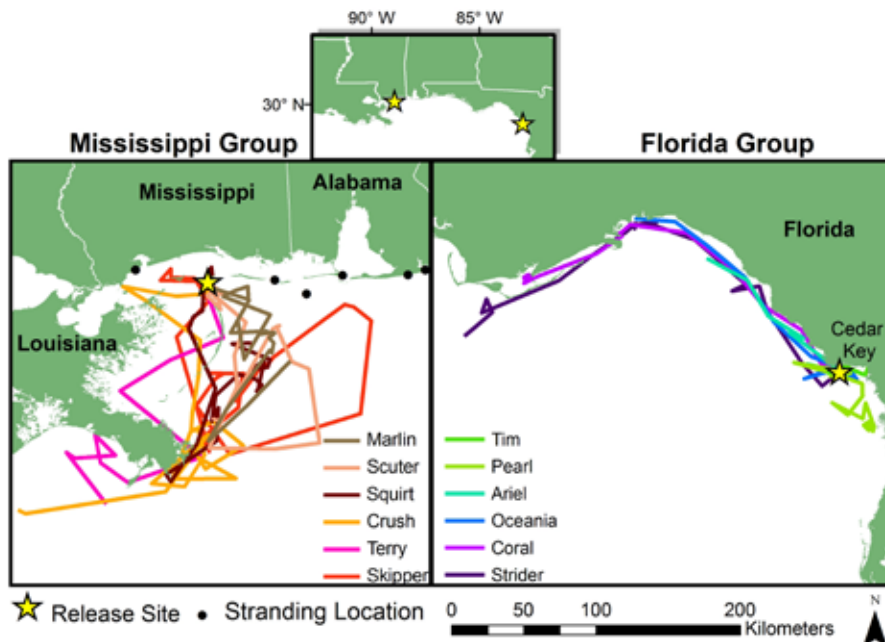
possibilities of statistical bias from the few turtles with exceptionally longer track durations.

Tracking paths for the turtles released in Mississippi indicated that they migrated to warmer waters offshore when water temperatures decreased; but they did not travel far. These individuals stayed in the general area of the Mississippi Sound and adjacent Louisiana waters during the 60-day tracking period, moving farther away from both their hooking/stranding location ( $r = 0.55$ ,  $p < 0.01$ ) and their release location ( $r = 0.48$ ,  $p < 0.01$ ) (Fig. 1). However, they did not travel farther than 183 km (Crush, 55.63 days after release) from their hooking/stranding locations within the 60-day period covered in this analysis.

In contrast, the majority of the turtles released in Florida did not remain in the area where they were initially released. Within days of the release, four out of six turtles quickly began swimming up the coastline toward Alabama and Mississippi, moving away from their release site ( $r = 0.58$ ,  $p < 0.01$ ) and closer to their hooking/stranding sites ( $r = -0.40$ ,  $p < 0.01$ ).

The slope of the regression line that best fits the data for the correlations between hours after release and distance from stranding sites were in opposite directions and almost twice as large for the Florida turtles (slope = -0.12) as for the Mississippi turtles (slope = 0.059). However, the average swimming speed was significantly faster for the Mississippi turtles (1.48 km/hr) than for the Florida turtles (1.16 km/hr;  $t(9) = 2.43$ ,  $p < 0.05$ ). This indicates that the Florida turtles were not moving as fast as the Mississippi turtles but were moving in a more direct line, in this case toward the hooking/stranding location, whereas the Mississippi turtles were moving generally away from their hooking/stranding site and not in a direct line.

The results indicated that the juvenile and subadult Kemp's ridleys released in Mississippi waters displayed a significant degree of site fidelity to the north central Gulf of Mexico. They stayed in the general area of Mississippi and Louisiana waters whereas several turtles that were released from Cedar Key, FL displayed western



**Figure 1.** Satellite tagging tracks of the turtles released in Mississippi (left) and Florida (right).

directional movements. These conclusions were supported by a home range analysis of the Mississippi-released turtles (Broadway *et al.* 2012 in prep), which detected a 100% utilization range from 5,570 to 12,134 km<sup>2</sup> (mean = 8,787 km<sup>2</sup> ± 2,294 SD) for individual turtles. They went no farther south than 28.7 °N during the winter months (Broadway *et al.* in prep). It is important to note that three of the four Florida-released turtles showing directional movements stopped transmitting before they reached their original hooking/stranding locations, and the fourth continued to its hooking/stranding location but did not spend considerable time there. Overall, the results of this study imply that it is best to release turtles near their hooking/stranding location when possible.

Interestingly, one of the Florida-released turtles, Strider, was tracked past its original hooking/stranding location to the vicinity of Rancho Nuevo, Mexico, which is the main nesting location for this species (Hildebrand 1963). Strider remained in this area for approximately two weeks in March before returning north to waters along the Texas/Louisiana border. Based on serum testosterone levels measured prior to release in April 2011 (0.846 ng/ml), and compared to typical levels (Rostal *et al.* 1998), Strider was determined to be male. Although Strider's carapace (curved notch-tip) was measured to be 54.5 cm at the time of release, which is lower than the widely accepted 60 cm threshold for categorizing Kemp's ridleys as mature, Gregory & Schmid (2001) suggested that maturation could occur prior to reaching this size. Shaver *et al.* (2005) showed that even though the majority of males reside near Rancho Nuevo year round some males can migrate away post-mating. Adult females have been tracked migrating from the Atlantic coast of Florida (Schmid 1995) and northern Gulf of Mexico (Renaud *et al.* 1996) to Rancho Nuevo, but the authors believe this is the first instance of tracking a newly mature Kemp's ridley male on its migration to mating grounds near the nesting beach.

Previous studies have examined the movements of immature Kemp's ridley sea turtles in other regions (Renaud & Williams 2005; Schmid *et al.* 2003; Seney & Landry 2011). Schmid *et al.*

(2003) tracked subadult Kemp's ridleys via radio and sonic telemetry in west central Florida to investigate home range sizes and habitat use. Turtles preferred to forage around rock outcroppings and in live benthic habitats, and several turtles displayed relatively small home ranges during the summer months (Schmid *et al.* 2003). Renaud & Williams (2005) tracked the movements of wild-caught and rehabilitated turtles in the northwestern Gulf of Mexico, Gulf coast of Florida and Atlantic seaboard from North Carolina to Florida. The majority of the monitored juvenile turtles remained within 15 km of their nearshore capture site and were characterized as habitat faithful. The authors also detected offshore movements as water temperatures cooled seasonally (Renaud & Williams 2005). More recently, Seney & Landry (2011) tracked rehabilitated immature Kemp's ridleys via satellite telemetry in the northwestern Gulf of Mexico and observed concentrated movements near tidal passes, fishing piers and within bay systems. The

conclusions of these studies correspond with the movements observed from the Mississippi-released turtles. These turtles seasonally migrated to offshore waters; however, five of the six were observed returning to the nearshore waters of the Mississippi Sound the next year (Broadway *et al.* in prep). The sixth turtle, Terry, stopped transmitting signals after only 23 days. Future analyses will examine summer movements and specific habitat use of juvenile Kemp's ridley sea turtles in the Mississippi Sound.

This is the first study to examine the movements of immature Kemp's ridley sea turtles via satellite telemetry in the north central Gulf of Mexico. Other satellite telemetry studies have provided insight into the use of poorly understood developmental grounds by juvenile loggerhead (Polovina *et al.* 2006) and juvenile green sea turtles (Hart & Fujisaki 2010). The revised recovery plan for the Kemp's ridley sea turtle (NMFS, USFWS & SEMARNAT 2011) calls for a better comprehension of habitat use of all life history stages. The north central Gulf of Mexico has been identified in the past to represent important developmental habitat for this species (Ogren 1989), yet data regarding habitat use and site fidelity are deficient. Additionally, as the Kemp's ridley population continues to recover, the chances for adverse human interactions, notably fishery interactions, may increase (Seney & Landry 2011). This potential has been underscored by the abnormally high number of Kemp's ridley strandings since 2010 (NOAA 2012), even though the cause(s) for this mortality is not fully understood. Therefore, more current data on the habitat use and movements of all life history stages of Kemp's ridley turtles will aid effective conservation and management throughout its range. The long-term study recently initiated by the Institute for Marine Mammal Studies will serve to fill this knowledge gap in the north central Gulf of Mexico and will contribute to the continued recovery of Kemp's ridleys.

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# Effects of Anthropogenic Activities on Sea Turtle Nesting Beaches along the Mombasa-Kilifi Shoreline, Kenya

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Sea turtles are an ancient lineage of reptiles with a fossil record of more than 200 million years (Pritchard 1979). Mature female sea turtles move periodically from marine to terrestrial habitats to lay eggs that incubate on oceanic beaches. During the reproductive period, both adults and eggs are vulnerable to over-exploitation by humans, habitat destruction, predation, artificial lighting, beach armoring, plastics and other marine debris (Pritchard 1979). Five species of marine turtles have been documented within Kenya waters (Frazier 1975): the green turtle (*Chelonia mydas* Linnaeus 1758), hawksbill turtle (*Eretmochelys imbricata* Linnaeus 1766), loggerhead turtle (*Caretta caretta* Linnaeus 1758), olive ridley turtle (*Lepidochelys olivacea* Eschscholtz 1829) and leatherback turtle (*Dermochelys coriacea* Vandelli 1761). Of these, green, hawksbill and olive ridley turtles are known to use Kenya beaches for reproductive activity (Okemwa *et al.* 2004).

Oceanic beaches are continuously changing as natural forces of wind and water meet land. These changes, which have been taking place for millions of years, are linked to variation in wind, waves, currents and sea level. It is not only natural forces that change beaches, however, human activities do as well, especially stone mining and sand harvesting, construction of buildings and walls close to the sea, and beach tourist activities (Nelson & Dickerson 1988). Mature marine female turtles nest on specific beaches, that is, their natal beach where they were incubated and hatched in the past (Bowen *et al.* 1989). Consequently, many mature females may fail to nest in suitable areas if their natal nesting habitats have been destroyed. Loss of suitable nesting habitats has been identified as one of the major threats facing the recovery of sea turtle populations in Kenya (UNEP 1998). Encroachment by manmade structures prevents the natural movement of sand, thus resulting in the degradation of beach habitats through factors such as beach erosion.

Much of the Kenya coastal zone has experienced rapid human population growth, spiraling demand for marine resources and unplanned development (UNEP 1998). Despite protection by the Kenya government (Fisheries Act 1989; and the Wildlife Management and Conservation Amendment Act 1989), illegal harvesting of adult nesting female turtles and egg collection are rampant (Olendo 1993).

In view of these anthropogenic activities and natural beach processes it is important to study the impact of beach development, tourism, and beach erosion and accretion on sea turtle nesting habitats. These activities and processes affect the biophysical characteristics of beaches, which may in turn affect the beaches' suitability for turtle nesting. Previous studies, (e.g., Horrocks & Scott 1991; Kamel & Mrosovsky 2004; Marcus & Maley 1987; Mortimer 1990), have shown that suitability for nesting by turtles depends on the following biophysical characteristics; beach vegetation, offshore approach, beach slope, beach width and beach material. This study

therefore determined how biophysical characteristics have changed in different locations at the beaches between 1986 and 2006, and in turn, how these changes have impacted the availability of potential suitable sites for nesting. The study was conducted on about 45 km of shoreline on the western Indian Ocean (Fig. 1) using field surveys and satellite imagery. Ground-truthing, i.e., the identification of features on the beach to aid the interpretation of satellite imagery was conducted from September 2006 to December 2006.

The nature of the offshore approach was characterized as the presence or absence of any obstructions such as rocks or strewn boulders from the low water mark to the start of the beach vegetation line shoreward. Presence of such obstructions may prevent marine turtle emergence onto the beach. The beaches were then classified as either having an open offshore approach or an obstructed approach (including partial obstruction). The study included partially obstructed with obstructed approaches so there were only two categories, i.e., clear/open and obstructed.

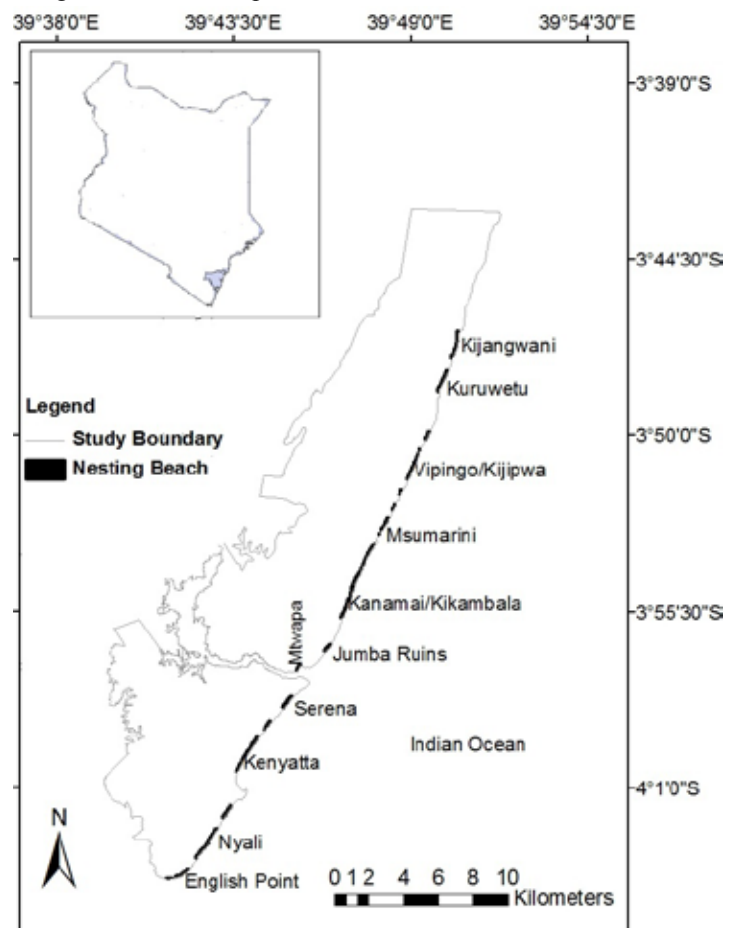


Figure 1. Location of marine turtle nesting beaches along the Mombasa-Kilifi shoreline in Kenya.



Beach width was determined as the horizontal distance between the low water mark (the start of the foreshore) and the start of the beach vegetation line. In the absence of vegetation, the distance was measured to the nearest beach structures at the landward edge of the sandy beach. We measured width using a tape measure, and recorded the presence or absence of vegetation on the nesting beaches.

The composition of beach substrate was determined from sand samples taken from all beaches. Three samples were collected from a random area of the beach, at nest depth using a soil auger. These samples were then thoroughly mixed to make a representative sample and a sub-sample of 200 g was taken. Granulometric analysis was then conducted on the sub-samples and a mean particle diameter for each beach was determined. The Wentworth scale of particle diameter (Pentthick 1984) was then used to determine the type of beach material at each beach. To determine the slope of the beach, the Wentworth scale for slope (Pentthick 1984) was used.

Existing nesting beaches were mapped using coordinates determined by a GPS during ground-truthing. These coordinates were then entered into ArcGIS 9.2 and polygons representing the nesting beaches were created. Existing nesting beaches were identified by the presence of nests observed during the field survey and information from key informants (fishermen and turtle conservation group members).

Potential suitable nesting sites in 1986, 2001 and 2006, were mapped by identifying sites that had 1) sandy beaches and open offshore approaches; 2) slopes greater than 4° and less than 12.5°; and 3) were approximately perpendicularly located 30 m away landwards from the shoreline. This identification was done from a combination of classified Landsat imagery and a GIS-generated slope map. Three separate map layers representing approach, slope and beach material were created. These maps were then combined to distinguish suitable and unsuitable areas.

Contours were digitized from a topographic map and a Digital Elevation Model (DEM) was created. A slope image derived from the DEM was then reclassified to create a Boolean image where a value of 1 was given to areas with slopes between 4° and 12.5°, and a value of 0 was assigned to all other areas. A third criterion map was created for areas 30 m landwards away from the shoreline by first digitizing the shoreline for the entire study area and creating a 30 m buffer. A Boolean image was created with areas 30 m landward away from the shoreline having a value of 1 and those less than 30 m being assigned a value of 0.

An overlay operation (multiply option) was then performed where the three Boolean criteria images were imposed on each other. The resulting image showed areas with suitable nesting beaches with a value of 1 and those beaches that were unsuitable habitats had a value of 0. This was done for both 1986 and 2001.

To determine the extent and pattern of change in the nesting beaches between 1986, 2001 and 2006, an overlay operation (subtraction option) was carried out. The 2001 map of suitable nesting beaches was overlaid on the 1986 map and again for 2006 compared to the 2001 map. The Area module in Idrisi 3.2 was used to calculate the changes in size of suitable nesting beaches.

A combination of GIS-based change detection methods and field surveys was used to link changes in land use and land cover characteristics to the changes in the sizes of nesting beaches. Field surveys were used to obtain complementary information on possible causes of land use and land cover in the study area. A total of 109

nests of three turtle species: green turtle, hawksbill turtle and olive ridley turtle were recorded (Fig. 2). These nests were observed on seven beaches that were being used for nesting during the study.

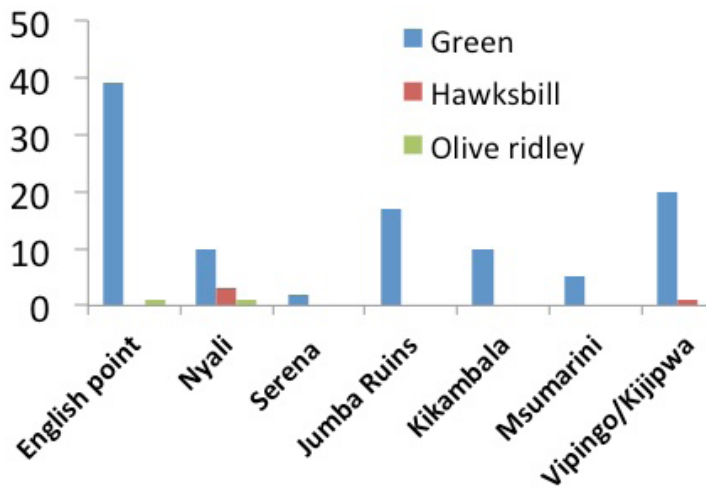
All beaches used for nesting were found to be composed of sandy surface material. No nesting activity was observed on beaches with muddy or rocky surfaces. Of the 109 nests observed, 59 were found on beaches with medium-grain sand, 26 nests on coarse sand and 24 on fine sand beaches. Of the seven beaches that were used for nesting, 3 had a medium-grain sand surface, 2 had fine sand and 2 had a coarse sand surface (Table 1).

Beach slopes that were used for nesting ranged from 5° to 9°, while beach width varied from 31 m to 60 m (Table 2). Most of the beaches used for nesting had open offshore approaches consisting of sand. A small section at English Point Beach, however, had obstacles mainly consisting of debris washed ashore and concrete blocks from destroyed sea walls. We also observed that a section between English Point and Nyali Beach had a sandy beach but it was not being used for nesting because there was a rocky approach.

Nests were mostly found where some vegetation was present. About 80% of the nests were found at the vegetation line while 10% of the nests were found on the open beach. The remaining 10% were

Beach	Sample	Grain size		
		(mm)	Mean	SD
English Point	1	0.012	0.012	0.003
	2	0.008		
	3	0.015		
Nyali	1	0.005	0.008	0.002
	2	0.010		
	3	0.009		
Serena	1	0.014	0.014	0.004
	2	0.010		
	3	0.019		
Jumba	1	0.006	0.006	0.001
	2	0.008		
	3	0.005		
Kikambala	1	0.049	0.037	0.01
	2	0.037		
	3	0.025		
Msumarini	1	0.045	0.049	0.01
	2	0.053		
	3	0.049		
Vipino/Kijipwa	1	0.065	0.057	0.015
	2	0.054		
	3	0.052		

**Table 1.** Mean sandy soil grain diameter for each beach.



**Figure 2.** Distribution of nests along the study beaches.

found within the vegetation. The most common vegetation present where the eggs were laid included vines (*Ipomoea pes-caprae*) and Marram grass (*Ammophila* spp).

In 1986, 11 beaches exhibited suitable characteristics (slope, beach width, approach and vegetation) for sea turtle nesting habitats. These beaches had a total combined area of 84.6 ha. In 2001, 10 beaches were found to be suitable for nesting, with a total combined area of 144.9 ha. This study, therefore, showed an increase in the area suitable for nesting between 1986 and 2001 by 60.3 ha. The area suitable for nesting in 2006 was only 41.7 ha. This area was spread across 7 beaches (Table 3) indicating a decline of 103.2 ha in suitable beaches for nesting between 2001 and 2006.

Eight land cover and land use classes were created from the two Landsat images (Table 4). Although some land classes that were inland, such as plantations, did not have any influence on nesting beaches, other land use changes such as built-up areas and thickets that were close to the beaches were found to have an influence on the suitability of nesting beaches. Increased beach development, especially on the Mombasa shoreline, was determined to be one of the major causes of altered nesting beach area, accounting for nearly 60% of the changes. Development, e.g., construction of sea walls within the Mombasa shoreline where most of the nests were located hindered accessibility by hatchlings to the water. Multiple turtle species have been found to nest on beaches with an open offshore approach. This was also observed for the three species nesting in the study area. Mortimer (1995) and Godley *et al.* (2001) concluded that

Beach	Beach slope	Beach width	Nests
English Point	7°	41	42
Nyali	5°	60	14
Serena	7°	52	2
Jumba Ruins	7°	48	17
Kanamai/Kikambala	5°	40	10
Msumarini	9°	31	5
Vipingo	9°	31	21

**Table 2.** Beach slope and width and the number of nests on each beach in the study area in 2006.

the primary reasons green and hawksbill turtles at Ascension Island avoided beaches with rock strewn approaches was because moving over such rocks was not only dangerous but also increased the threat of predation due to obstructed movement. All three turtle species in the study area were found to nest on sandy surface beaches. Kamel *et al.* (2004) observed olive ridley turtles nesting on muddy shores; however, no nests were observed in this study despite the availability of muddy shoreline habitat at Mtwapa beach. This study did not establish the influence of sand particle size on the three observed species, thus confirming the findings of Mortimer (1990) that beach sand types were less important than slope and offshore configuration of the beach. Most nests in the study were found on beaches with steep slopes. Fish *et al.* (2005) determined that in Bonaire, Dutch Caribbean Antilles, nesting density increased with beach slope for the greens, hawksbills, loggerheads, olive ridleys and leatherbacks.

Despite having a short beach width, Vipingo and Msumarini beaches still had nesting activity due to their steep gradient. This concurred with the findings of both Horrocks *et al.* (1991), and Weishampel *et al.* (2003), who observed that beach width and gradient were crucial for nest placement. In this study, short beaches with a steep gradient had nesting activity because they were safe from being inundated by seawater. Of the nests in this study, 72% were found in areas where vegetation was present. The results, therefore, concurred with those of Bustard (1972) and Mortimer (1990) who found that the presence of vegetation at nesting sites was important. These studies both showed that hatchling emergence success at beaches with loose sand was low as the nests tended to crumble and suffocate some of the hatchlings. This did not happen when sand was held together by vegetation.

This study concludes that there were changes in the area of suitable nesting beaches along the Mombasa-Kilifi coastline between 1986 and 2006. Some of the beaches had completely been abandoned by nesting turtles due to habitat loss and alteration while other beaches had either decreased or increased in size (Table 3). English Point was one of the beaches with notable changes despite exhibiting the highest nesting density. This beach, located on the Mombasa shoreline, is fronted by cliffs and residential areas. By the year 2006, six nesting sites had been abandoned in this location

Beach	1986	2001	2006
English Point	4.0	-	0.4
Nyali	14.7	35.2	0.6
Kenyatta	10.6	21.1	-
Serena	2.3	1.3	4.2
Mtwapa	1.8	4.1	-
Jumba Ruins	3.1	7.3	2.4
Kanamai/Kikambala	16.6	32.0	16.3
Msumarini	12.6	14.0	13.4
Vipingo/Kijipwa	10.8	18.6	4.4
Kuruwetu	1.1	2.3	-
Kijangwani	7.0	9.0	-
Total	84.6	144.9	41.7

**Table 3.** Size (ha) of nesting beaches suitable for nesting in 1986, 2001 and 2006.

Land use/land cover	1986	2001	Change
Built up	419.66	870.33	450.67
Thickets	7,098.56	6,523.99	-574.57
Forest	572.34	95.92	-476.42
Agricultural land	955.26	1063.18	107.92
Plantations	5,907.47	5,272.95	-634.52
Bare surface	312.76	130.9	-181.86
Mudflats	176.87	182.85	5.98
Rangelands	10,610.87	11,115.47	504.60
Others	102.25	900.40	798.15
Total	26,155.99	26,155.99	0

**Table 4.** Land use and land cover changes (ha) between 1986 and 2001.

due to encroachment by residential buildings and the construction of sea walls around the sites.

Other beaches, such as Kenyatta, Nyali and Serena were also affected by the encroachment of human activities. Tourist hotels and tourist activities (e.g., beach football and volleyball) were prevalent on these beaches. In Kenyatta and Serena beaches, some tourist hotels have completely encroached upon beaches that had previously been used for nesting through construction of sea walls and the use of beach chairs.

Other activities that affected the nesting beaches included the removal of beach front vegetation and the planting of exotic vegetation, such as the Australian pine (*Casuarina equisetifolia*). The increased presence of the Australian pine on the Kenya coastline has been attributed to efforts to control beach erosion (UNEP 1998), but it might be detrimental to sea turtle nesting; this has been observed by Marcus *et al.* (1987) in South Florida where Australian pine roots prevented nesting turtles from building good nests.

The land use and land cover change analysis showed an increase in the size of exclusive expansive high cost residential areas on the Mombasa shoreline where most of the beaches were found to have been affected by structures such as sea walls and buildings. Also, in these areas the vegetation present near the beaches had been cleared, thus leaving the nesting sites exposed. Conversely, beaches along Kilifi's shoreline such as Jumba Ruins, Msumarini and Vipingo did not have large areas of encroachment. Most of these beaches were bordered by cliffs and thickets, which seemed to deter any encroachment. Nesting beaches in both the Mombasa and Kilifi shorelines that were bordered by cliffs were the least affected by development. The beaches in Kilifi seemed to be primarily negatively affected by erosion caused by wave action.

At Jumba Ruins, the lack of encroachment by human activities was likely due to the fact that most of the beach was fronted by the Jumba Ruins prehistoric site. This site is managed by the National Museums of Kenya, which has protected it from any form of human activities. Mtwapa beach experienced some unique changes during the study period. Despite having suitable nesting habitat, the approach from the sea at Mtwapa consisted mainly of muddy shores with dense mangrove vegetation. The land use and land cover change analysis showed an increase in the area occupied by muddy shores over the study period in this area. Olive ridley turtles did not use this beach even though they are known to use beaches with muddy

shores for nesting (Pritchard 1979). This is likely due to the presence of dense mangrove vegetation that hindered access to the beach.

The increase in size of suitable nesting areas especially between 1986 and 2001, and in the English Point region between 2001 and 2006 may be attributed to the natural beach processes of erosion and accretion. The areas with sandy beaches could have varied between 1986 and 2001 in the amount of sand probably because the two satellite images despite being taken during the same season could have been taken when erosion or accretion had occurred. It was difficult during this study to ascertain exactly the time of the year when these two processes actually occurred in 1986 and 2001 due to lack of information. Although the sandy beach areas available in 1986 and 2001 varied due to either accretion or erosion, the processes could have been influenced and accelerated by human activities like beach armoring. At the English Point beach for example, the presence of sea walls and sand bags placed on the beaches indicated that wave erosion was affecting residences and that action was being taken to further prevent erosion.

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## From Suriname to Ceará. Green Turtle Found Dead on the Coast of Ceará, Brazil

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On 16 May 2012, an adult female green turtle (*Chelonia mydas*) was found dead on the beach at -3.8253°S, -38.4002°W in Ceará, Brazil; it had a Monel flipper tag on its left front flipper (SUR0000032). This individual had been tagged during the evening of 16 May 2010 on Matapica Beach, Suriname (5.9945°N, -54.9845°W), after she was observed nesting. The distance between where she was tagged and where she was found stranded was nearly 2,000 km. At the time of tagging, her curved carapace length (CCL) measured 103 cm. When found in Ceará, the CCL was 111 cm. When discovered, the carcass was moderately decomposed, and this may have affected the carapace length measurements to a small degree.

The coastal waters off of the state of Ceará have long been documented as a foraging area for juvenile and adult (27 - 132 cm CCL) green turtles (Ferreira 1968; Lima *et al.* 2003; Marcovaldi *et al.* 2001). Observations of flipper tag returns in Ceará have shown that many of the green turtles found in this area come from a variety of foraging and nesting populations, including Suriname and the Guianas (Pritchard 1973; Schulz 1975), Ascension Island (Carr *et al.* 1964), Costa Rica and Puerto Rico (Lima *et al.* 2008). Genetic analyses of juvenile and adult turtles found incidentally captured or stranded in Ceará demonstrate that the Surinam nesting population is a major source rookery for that green turtle foraging area (Bowen *et al.* 1992; Naro-Maciel *et al.* 2007). However, this is the first tag recovery in Brazil of an adult green turtle from Suriname in several decades, although the tagging effort in Suriname has not been consistent over this time frame.

This tag recovery is concordant with complementary datasets derived from flipper tags and genetic mixed stock analyses (Lima *et al.* 2008; Naro-Maciel *et al.* 2007; Pritchard 1973). Data from a satellite tracking study of green turtles in Ceará showed that juveniles and adults (41 - 116 cm CCL) generally remained near the Ceará coast, with a few individuals moving several hundred kilometers away from the initial release point. However, the average time that the turtles were tracked was 70 days (Godley *et al.* 2003).

Thus, there is a lack of information on the movements of adult and juvenile green turtles in this region over longer time and geographic scales. We recommend that more satellite tags be deployed on juvenile and adult green turtles foraging in Ceará to fully describe their movements. We also recommend that flipper tags continue to be placed on both adult and juvenile green turtles in the wider Central Western Atlantic region as a cost-effective means to elucidate turtle movements between different habitats.

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## New Northern Limit of Nesting of *Lepidochelys olivacea* in the East Atlantic Ocean: North Senegal (West Africa)

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The Langue de Barbarie National Park in Senegal was created in January 1976. It is a long dune strip about 15 km in length and 1 km wide, between the Senegal River and the sea. Green turtles (*Chelonia mydas*) nest sporadically on this long beach (Fretey 1990). On the morning of 21 June, 2011, a fisherman discovered tracks of a turtle that he did not recognize on the beach within the national park (15.986944 °N, -16.513611 °W). The guards of the park were informed, and they left the nest in situ with a wire fence protecting it. Fifty-five days later, 112 hatchlings emerged, and the hatchlings were identified by one of us as *Lepidochelys olivacea* (Eschscholtz, 1829). Overall success rate of this nest was 98% (out of 117 total eggs laid). This case marks the northernmost known for this species in East Atlantic Ocean

The distribution and nesting of *Lepidochelys olivacea* in West Africa *sensu stricto* are still poorly known (Fretey 2001; Varo-Cruz *et al.* 2011). Maigret (1983) estimated that the northern limit of this species in the East Atlantic was Senegal. However, the collection of the Institut Fondamental d'Afrique Noire (IFAN) in Dakar has a ridley turtle (accession number 55.32.136) collected in 1955 from Port-Étienne (now called Nouadhibou) in Mauritania (nearly 21 °N). Interestingly, Carr (1957) reported that the most northerly limit for olive ridleys in the East Atlantic Ocean was 21 °N. Arvy & Dia (1997) reported the existence of olive ridley carapaces in Mauritania, and recently, news cases of olive ridleys were observed South of the capital Nouakchott (Mint Hama *et al.*, in press). To date, there has been no confirmation of nesting by olive ridleys in Mauritania.

In terms of nesting in Senegal, Cadenat (1949) reported the capture of two immature olive ridley turtles in nets sharks off of Hann beach in Senegal, and suggested that they could have originated from nests laid on beaches near Dakar, Gorée, N'Gapara and Joal (Cadenat, 1957).

The Sea Museum of Gorée Island holds in its collection one olive ridley hatchling it remains unclear whether it came from nest laid on a Senegalese beach (Fretey 1998). Maigret (1977) posited that olive ridleys may possibly nest on the coasts of Senegal, and a 1982 report of the Direction of the National Parks of Senegal suggested that the species is likely to reproduce within the limits of protected

area of the Delta of Saloum. However, the record we report here is the first confirmed nest of olive ridleys in Senegal.

Within the central Atlantic Africa region, there is confirmed nesting of olive ridleys on several islands in Guinea-Bissau. The highest density nesting site in the region appears to be on the islands of Orango Large, Imbone, Adonga and Orangozinho in Orango National Park of the Archipelago of Bijagos (Cetry *et al.* 2009). Fretey & Malaussena (1991) noted olive ridleys nesting further south, in Sierra Leone, on the small island of Baki, within the archipelago of Sherbro – Turtle Islands. We recommend that sea turtle researchers in the region be vigilant in observing and reporting the occurrences of this species, to help illuminate more precisely their nesting and foraging distribution.

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## THESES & DISSERTATIONS

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